Chapter 5

Receiving Water and Other Impacts

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Desired Water Uses Versus Stormwater Impacts

The main purpose of treating stormwater is to reduce its adverse impacts on receiving water beneficial uses. Therefore, this report on wet-weather flow management systems includes an assessment of the detrimental effects that runoff is actually having on a receiving water.

Urban receiving waters may have many beneficial use goals, including:

1. Stormwater conveyance (flood prevention).
2. Biological uses (e.g., warm water fishery, biological integrity).
3. Non-contact recreation (e.g., linear parks, aesthetics, boating).
4. Contact recreation (swimming).
5. Water supply and irrigation.

With full development in an urban watershed and with no stormwater controls, it is unlikely that any of these uses can be obtained. With less development and with the application of stormwater controls, some uses may be possible. Unreasonable expectations should not be placed on urban waters, because the cost to obtain these uses may be prohibitive. With full-scale development and lack of adequate stormwater controls, severely degraded streams will be common.

Stormwater conveyance and aesthetics should be the basic beneficial use goals for all urban waters. Biological integrity should also be a goal, but with the realization that the natural stream ecosystem will be severely modified with urbanization. Certain basic controls, installed at the time of development, plus protection of stream habitat, may enable partial realization of some of these basic goals in urbanized watersheds. Careful planning and optimal utilization of stormwater controls are necessary to obtain these basic goals in most watersheds. Water contact recreation, consumptive fisheries, and water supplies are not appropriate goals for most urbanized watersheds. These higher uses may be possible in urban areas where the receiving waters are large and drain mostly undeveloped areas.

In general, monitoring of urban stormwater runoff has indicated that the biological beneficial uses of urban receiving waters are most likely affected by habitat destruction and long-term pollutant exposures (especially to macroinvertebrates via contaminated sediment). Documented effects associated from acute exposures of toxicants in the water column are rare (Field and Pitt 1990, Pitt 1995).

Receiving water pollutant concentrations resulting from runoff events and typical laboratory bioassay test results have not indicated many significant short-term receiving water problems. As an example, Lee and Jones-Lee (1993) state that exceedences of numeric criteria by short-term discharges do not necessarily imply that a beneficial use impairment exists. Many toxicologists and water quality experts have concluded that the relatively short periods of exposures to the toxicant concentrations in stormwater are not sufficient to produce the receiving water effects that are evident in urban receiving waters, especially considering the relatively large portion of the toxicants that are associated with particulates (Lee and Jones-Lee 1995a and 1995b). Lee and Jones-Lee (1995a and 1995b) conclude that the biological problems evident in urban receiving waters due to stormwater discharges are mostly associated with illegal discharges and that the sediment bound toxicants are of little risk. Mancini and Plummer (1986) have long been advocates of numeric water quality standards for stormwater that reflect the partitioning of the toxicants and the short periods of exposure during rains. Unfortunately, this approach attempts to isolate individual runoff events and does not consider the accumulative adverse effects caused by the frequent exposures of receiving water organisms to stormwater (Davies 1995, Herricks 1995 and Herricks et al. 1996). Recent investigations have identified acute toxicity problems associated with short-term (about 10 to 20 day) exposures to adverse toxicant concentrations in urban receiving streams (Crunkilton et al. 1997). However, the most severe receiving water problems are likely associated with chronic exposures to contaminated sediment and to habitat destruction.

The effects of stormwater on receiving waters are very site specific. Accordingly, site investigations of local waters are highly recommended to understand the magnitude and like cause of the problems. Burton and Pitt (1996) have prepared a book that details site investigation procedures that can be used for local waters. The following is a summary of recent work describing the toxicological and ecological effects of stormwater.

Toxicological Effects of Stormwater

The need for endpoints for toxicological assessments using multiple stressors was discussed by Marcy and Gerritsen (1996). They used five watershed-level ecological risk assessments to develop appropriate endpoints based on specific project objectives. Dyer and White (1996) also examined the problem of multiple stressors affecting toxicity assessments. They felt that field surveys rarely can be used to verify simple single parameter laboratory experiments. They developed a watershed approach integrating numerous databases in conjunction with in-situ biological observations to help examine the effects of many possible causative factors. Toxic effect endpoints are additive for compounds having the same "mode of toxic action", enabling predictions of complex chemical mixtures in water, as reported by Environmental Science & Technology (1996a). According to EPA researchers at the Environmental Research Laboratory in Duluth, MN, there are about five or six major action groups that contain almost all of the compounds of interest in the aquatic environment. Much work still needs to be done, but these new developing tools may enable improved prediction of in-stream toxic effects of stormwater.

Ireland et al. (1996) found that exposure to ultraviolet (UV) radiation (natural sunlight) increased the toxicity of PAH contaminated urban sediments to C. dubia. The toxicity was removed when the UV wavelengths did not penetrate the water column to the exposed organisms. Toxicity was also reduced significantly in the presence of UV when the organic fraction of the stormwater was removed. Photo-induced toxicity occurred frequently during low flow conditions and wet weather runoff and was reduced during turbid conditions.

Johnson et al. (1996) and Herricks et al. (1996) describe a structured tier testing protocol to assess both short-term and long-term wet weather discharge toxicity that they developed and tested. The protocol recognizes that the test systems must be appropriate to the time-scale of exposure during the discharge. Therefore, three time-scale protocols were developed, for intra-event, event, and long-term exposures. The use of standard whole effluent toxicity (WET) tests were found to over-estimate the potential toxicity of stormwater discharges.

The effects of stormwater on Lincoln Creek, near Milwaukee, WI, were described by Crunkilton et al. (1997). Lincoln Creek drains a heavily urbanized watershed of 19 mi^2 that is about nine miles long. On-site toxicity testing was conducted with side-stream flow-through aquaria using fathead minnows, plus in-stream biological assessments, along with water and sediment chemical measurements. In the basic tests, Lincoln Creek water was continuously pumped through the test tanks, reflecting the natural changes in water quality during both dry and wet weather conditions. The continuous flow-through mortality tests indicated no toxicity until after about 14 days of exposure, with more than 80% mortality after about 25 days, indicating that short-term toxicity tests likely underestimate stormwater toxicity. The biological and physical habitat assessments supported a definitive relationship between degraded stream ecology and urban runoff.

Rainbow (1996) presented a detailed overview of heavy metals in aquatic invertebrates. He concluded that the presence of a metal in an organism couldn't tell us directly whether that metal is poisoning the organism. However, if compared to concentrations in a suite of well-researched biomonitors, it is possible to determine if the accumulated concentrations are atypically high, with a possibility that toxic effects may be present. Allen (1996) also presented an overview of metal contaminated aquatic sediments. Allen's book presents many topics that would enable the user to better interpret measured heavy metal concentrations in urban stream sediments.

Ecological Effects of Stormwater

A number of comprehensive and long-term studies of biological beneficial uses in areas not affected by conventional point source discharges have typically shown impairments caused by urban runoff. The following paragraphs briefly describe a variety of such studies.

Klein (1979) studied 27 small watersheds having similar physical characteristics, but having varying land uses, in the Piedmont region of Maryland. During an initial phase of the study, they found definite relationships between water quality and land use. Subsequent study phases examined aquatic life relationships in the watersheds. The principal finding was that stream aquatic life problems were first identified with watersheds having imperviousness areas comprising at least 12 percent of the watershed. Severe problems were noted after the imperviousness quantities reached 30 percent.

Receiving water impact studies were also conducted in North Carolina (Lenet et al. 1979, Lenet and Eagleson 1981, Lenet et al. 1981). The benthic fauna occurred mainly on rocks. As sedimentation increased, the amount of exposed rocks decreased, with a decreasing density of benthic macroinvertebrates. Data from 1978 and 1979 in five cities showed that urban streams were grossly polluted by a combination of toxicants and sediment. Chemical analyses, without biological analyses, would have underestimated the severity of the problems because the water column quality varied rapidly, while the major problems were associated with sediment quality and effects on macroinvertebrates. Macroinvertebrate diversities were severely reduced in the urban streams, compared to the control streams. The biotic indices indicated very poor conditions for all urban streams. Occasionally, high populations of pollutant tolerant organisms were found in the urban streams, but would abruptly disappear before subsequent sampling efforts. This was probably caused by intermittent discharges of spills or illegal dumpings of toxicants. Although the cities studied were located in different geographic areas of North Carolina, the results were remarkably uniform.

During the Coyote Creek, San Jose, CA, receiving water study, 41 stations were sampled in both urban and nonurban perennial flow stretches of the creek over three years. Short and long-term sampling techniques were used to evaluate the effects of urban runoff on water quality, sediment properties, fish, macroinvertebrates, attached algae, and rooted aquatic vegetation (Pitt and Bozeman 1982). These investigations found distinct differences in the taxonomic composition and relative abundance of the aquatic biota present. The non-urban sections of the creek supported a comparatively diverse assemblage of aquatic organisms including an abundance of native fishes and numerous benthic macroinvertebrate taxa. In contrast, however, the urban portions of the creek (less than 5% urbanized) affected only by urban runoff discharges and not industrial or municipal discharges, had an aquatic community generally lacking in diversity and was dominated by pollution-tolerant organisms such as mosquitofish and tubificid worms.

A major nonpoint runoff receiving water impact research program was conducted in Georgia (Cook et al. 1983). Several groups of researchers examined streams in major areas of the state. Benke et al. (1981) studied 21 stream ecosystems near Atlanta having watersheds of one to three square miles each and land uses ranging from 0 to 98% urbanization. They measured stream water quality but found little relationship between water quality and degree of urbanization. The water quality parameters also did not identify a major degree of pollution. In contrast, there were major correlations between urbanization and the number of species found. They had problems applying diversity indices to their study because the individual organisms varied greatly in size (biomass).

CTA (1983) also examined receiving water aquatic biota impacts associated with urban runoff sources in Georgia. They studied habitat composition, water quality, macroinvertebrates, periphyton, fish, and toxicant concentrations in the water, sediment, and fish. They found that the impacts of land use were the greatest in the urban basins. Beneficial uses were impaired or denied in all three urban basins studied. Fish were absent in two of the basins and severely restricted in the third. The native macroinvertebrates were replaced with pollution tolerant organisms. The periphyton in the urban streams were very different from those found in the control streams and were dominated by species known to create taste and odor problems.

Pratt et al. (1981) used basket artificial substrates to compare benthic population trends along urban and nonurban areas of the Green River in Massachusetts. The benthic community became increasing disrupted as urbanization increased. The problems were not only associated with times of heavy rain, but seemed to be affected at all times. The stress was greatest during summer low flow periods and was probably localized near the stream bed. They concluded that the high degree of correspondence between the known sources of urban runoff and the observed effects on the benthic community was a forceful argument that urban runoff was the causal agent of the disruption observed.

Cedar swamps in the New Jersey Pine Barrens were studied by Ehrenfeld and Schneider (1983). They examined nineteen wetlands subjected to varying amounts of urbanization. Typical plant species were lost and replaced by weeds and exotic plants in urban runoff affected wetlands. Increased uptakes of phosphorus and lead in the plants were found. The researchers concluded that the presence of stormwater runoff to the cedar swamps caused marked changes in community structure, vegetation dynamics, and plant tissue element concentrations.

Medeiros and Coler (1982) and Medeiros et al. (1984) used a combination of laboratory and field studies to investigate the effects of urban runoff on fathead minnows. Hatchability, survival, and growth were assessed in the laboratory in flow-through and static bioassay tests. Growth was reduced to one half of the control growth rates at 60% dilutions of urban runoff. The observed effects were believed to be associated with a combination of toxicants.

The University of Washington (Pederson 1981, Richey et al. 1981, Perkins 1982, Richey 1982, Scott et al. 1982, Ebbert et al. 1983, Pitt and Bissonnette 1983, and Prych and Ebbert undated) conducted a series of studies to contrast the biological and chemical conditions in urban Kelsey Creek with rural Bear Creek in Bellevue, WA. The urban creek was significantly degraded when compared to the rural creek, but still supported a productive, but limited and unhealthy salmonid fishery. Many of the fish in the urban creek, however, had respiratory anomalies. The urban creek was not grossly polluted, but flooding from urban developments had increased dramatically in recent years. These increased flows markedly changed the urban stream's channel by causing unstable conditions with increased stream bed movement, and by altering the availability of food for the aquatic organisms. The aquatic organisms were very dependent on the few relatively undisturbed reaches. Dissolved oxygen concentrations in the sediments depressed embryo salmon survival in the urban creek. Various organic and metallic priority pollutants were discharged to the urban creek, but most of them were apparently carried through the creek system by the high storm flows to Lake Washington. The urbanized Kelsey Creek also had higher water temperatures (probably due to reduced shading) than Bear Creek. This probably caused the faster fish growth in Kelsey Creek.

The fish population in the urbanized Kelsey Creek had adapted to its degrading environment by shifting the species composition from coho salmon to less sensitive cutthroat trout and by making extensive use of less disturbed refuge areas. Studies of damaged gills found that up to three-fourths of the fish in Kelsey Creek were affected with respiratory anomalies, while no cutthroat trout and only two of the coho salmon sampled in the forested Bear Creek had damaged gills. Massive fish kills in Kelsey Creek and its tributaries were also observed on several occasions during the project due to the dumping of toxic materials down the storm drains.

There were also significant differences in the numbers and types of benthic organisms found in urban and forested creeks during the Bellevue research. Mayflies, stoneflies, caddisflies, and beetles were rarely observed in the urban Kelsey Creek, but were quite abundant in the forested Bear Creek. These organisms are commonly regarded as sensitive indicators of environmental degradation. One example of degraded conditions in Kelsey Creek was shown by a specie of clams (Unionidae) that was not found in Kelsey Creek, but was commonly found in Bear Creek. These clams are very sensitive to heavy siltation and unstable sediments. Empty clam shells, however, were found buried in the Kelsey Creek sediments indicating their previous presence in the creek and their inability to adjust to the changing conditions. The benthic organism composition in Kelsey Creek varied radically with time and place while the organisms were much more stable in Bear Creek.

Urban runoff impact studies were conducted in the Hillsborough River near Tampa Bay, FL, as part of the U.S. EPA’s Nationwide Urban Runoff Program (NURP) (Mote Marine Laboratory 1984). Plants, animals, sediment, and water quality were all studied in the field and supplemented by laboratory bioassay tests. Effects of salt water intrusion and urban runoff were both measured because of the estuarine environment. During wet weather, freshwater species were found closer to Tampa Bay than during dry weather.

In coastal areas, these additional natural factors made it even more difficult to identify the cause and effect relationships for aquatic life problems. During another NURP project, Striegl (1985) found that the effects of accumulated pollutants in Lake Ellyn (Glen Ellyn, IL) inhibited desirable benthic invertebrates and fish and increased undesirable phyotoplankton blooms.

The number of benthic organism taxa in Shabakunk Creek in Mercer County, NJ, declined from 13 in relatively undeveloped areas to four below heavily urbanized areas (Garie and McIntosh 1986 and 1990). Periphyton samples were also analyzed for heavy metals with significantly higher metal concentrations found below the heavily urbanized area than above.

Many of the above noted biological effects associated with urban runoff are likely caused by polluted sediments and benthic organism impacts. Examples of heavy metal and nutrient accumulations in sediments are numerous. In addition to the studies noted above, DePinto et al. (1980) found that the cadmium content of river sediments can be more than 1,000 times greater than the overlying water concentrations and the accumulation factors in sediments are closely correlated with sediment organic content. Another comprehensive study on polluted sediment was conducted by Wilber and Hunter (1980) along the Saddle River in New Jersey where they found significant increases in sediment contamination with increasing urbanization.

The effects of urban runoff on receiving water aquatic organisms or other beneficial uses is very site specific. Different land development practices create substantially different runoff flow characteristics. Different rain patterns cause different particulate washoff, transport and dilution conditions. Local attitudes also define specific beneficial uses and, therefore, current problems. There are also a wide variety of water types receiving urban runoff and these waters all have watersheds that are urbanized to various degrees. Therefore, it is not surprising that urban runoff effects, though generally dramatic, are also quite variable and site specific.

Claytor (1996a) summarized the approach developed by the Center for Watershed Protection as part of their EPA sponsored research on stormwater indicators (Claytor and Brown 1996). The 26 stormwater indicators used for assessing receiving water conditions were divided into six broad categories: water quality, physical/hydrological, biological, social, programmatic, and site. These were presented as tools to measure stress (impacting receiving waters), to assess the resource itself, and to indicate stormwater control program implementation effectiveness. The biological communities in Delaware’s Piedmont streams have been severely impacted by stormwater, after the extent of imperviousness in the watersheds exceeds about 8 to 15%, according to a review article by Claytor (1996b). If just conventional water quality measures are used, almost all (87%) of the state’s non-tidal streams supported their designated biological uses. However, when biological assessments are included, only 13% of the streams were satisfactory.

Changes in physical stream channel characteristics can have a significant effect on the biological health of the stream. Schueler (1996) stated that channel geometry stability can be a good indicator of the effectiveness of stormwater control practices. He also found that once a watershed area has more than about 10 to 15% effective impervious cover, noticeable changes in channel morphology occur, along with quantifiable impacts on water quality and biological conditions.

Stephenson (1996) studied changes in streamflow volumes in South Africa during urbanization. He found increased stormwater runoff, decreases in the groundwater table, and dramatically decreased times of concentration. The peak flow rates increased by about two-fold, about half caused by increased pavement (in an area having only about 5% effective impervious cover), with the remainder caused by decreased times of concentration.

Fate of Stormwater Pollutants in Surface Waters

Many processes may affect urban runoff pollutants after discharge. Sedimentation in the receiving water is the most common fate mechanism because many of the pollutants investigated are mostly associated with settleable particulate matter and have relatively low filterable concentration components. Exceptions include zinc and 1,3-dichlorobenzene, which are mostly associated with the filtered sample portions.

Particulate reduction can occur in many stormwater runoff and CSO control facilities, including (but not limited to) catchbasins, swirl concentrators, fine mesh screens, sand or other filters, drainage systems, and detention ponds. These control facilities (with the possible exception of drainage systems) allow reduction of the accumulated polluted sediment for final disposal in an appropriate manner. Uncontrolled sedimentation will occur in relatively quiescent receiving waters, such as lakes, reservoirs, or slow moving rivers or streams. In these cases, the wide dispersal of the contaminated sediment is difficult to remove and can cause significant detrimental effects on biological processes.

Biological or chemical degradation of the sediment toxicants may occur in the typically anaerobic environment of the sediment, but the degradation is quite slow for many of the pollutants. Degradation by photochemical reaction and volatilization (evaporation) of the soluble pollutants may also occur, especially when these pollutants are near the surface of aerated waters (Callahan et al. 1979, Parmer 1993). Increased turbulence and aeration encourages these degradation processes, which in turn may significantly reduce toxicant concentrations. In contrast, quiescent waters would encourage sedimentation that would also reduce water column toxicant concentrations, but increase sediment toxicant concentrations. Metal precipitation and sorption of pollutants onto suspended solids increases the sedimentation and/or floatation potential of the pollutants and also encourages more efficient bonding of the pollutants to soil particles, preventing their leaching to surrounding waters.

Receiving waters have a natural capacity to treat and/or assimilate polluted discharges. This capacity will be exceeded sooner (assuming equal inputs), resulting in more degradation, in smaller urban creeks and streams, than in larger receiving waters. Larger receiving waters may still have ecosystem problems from the long-term build up of toxicants in the sediment and repeated exposures to high flowrates, but these problems will be harder to identify using chemical analyses of the water alone, because of increased dilution (Pitt and Bissonnette 1983).

In-stream receiving water investigations of urban runoff effects need a multi-tiered monitoring approach, including habitat evaluations, water and sediment quality monitoring, flow monitoring, and biological investigations, conducted over long periods of time (Pitt 1991). In-stream taxonomic (biological community structure) investigations are needed to help identify actual toxicity problems. Laboratory bioassay tests can be useful to determine the major sources of toxicants and to investigate toxicity reduction through treatment, but they are not a substitute for actual in-stream investigations of receiving water effects. In order to identify the sources and treatability of the problem pollutants, detailed watershed investigations are needed, including both dry and wet weather urban drainage monitoring and source area monitoring.

An estimate of the actual pollutant loads (calculated from the runoff volumes and pollutant concentrations) from different watershed areas is needed for the selection and design of most treatment devices. Several characteristics of a source area are significant influences on the pollutant concentrations and stormwater runoff volumes. The washoff of debris, soil, and pollutants depends on the intensity of the rain, the properties of the material removed, and the surface characteristics where the material resides. The potential mass of pollutants available to be washed off will be directly related to the time interval between runoff events during which the pollutants can accumulate.

Human Health Effects of Stormwater

Water Environment & Technology (1996b) reported on an epidemiology study conducted at Santa Monica Bay, CA, that found that swimmers who swam in front of stormwater outfalls were 50% more likely to develop a variety of symptoms than those who swam 400 m from the same outfalls (Haile et al. 1996). This was a follow-up study after previous investigations found that human fecal waste was present in the stormwater collection systems. Environmental Science & Technology (1996b) also reported on this Santa Monica Bay study. They reported that more than 1% of the swimmers who swam in front of the outfalls were affected by fevers, chills, ear discharges, vomiting and coughing, based on surveys of more than 15,000 swimmers. The health effects were also more common for swimmers who were exposed on days when viruses were found in the outfall water samples.

Water Environment & Technology (1996a) reported that the fecal coliform counts decreased from about 500 counts/100 ml to about 150 counts/100 ml in the Mississippi River after the sewer separation program in the Minneapolis and St. Paul area of Minnesota. Combined sewers in 8,500 ha were separated during this 10-year, $332 million program.

Groundwater Impacts from Stormwater Infiltration

Prior to urbanization, groundwater recharge results from infiltration of precipitation through pervious surfaces, including grasslands and woods. This infiltrating water is relatively uncontaminated. With urbanization, the permeable soil surface area through which recharge by infiltration could occur is reduced. This results in much less groundwater recharge and greatly increased surface runoff. In addition, the waters available for recharge generally carry increased quantities of pollutants. With urbanization, new problematic sources of groundwater recharge also occur, including recharge from domestic septic tanks, percolation basins and industrial waste injection wells, and from agricultural and residential irrigation.

The following paragraphs (from Pitt et al. 1994 and 1996) describe the stormwater pollutants that have the greatest potential of adversely affecting groundwater quality during inadvertent or intentional stormwater infiltration. Also included are suggestions on ways to minimize these potential problems.

Constituents of Concern

Nutrients

Nitrates are one of the most frequently encountered contaminants in groundwater. Groundwater contamination of phosphorus has not been as widespread, or as severe, as for nitrogen compounds. Whenever nitrogen-containing compounds come into contact with soil, a potential for nitrate leaching into groundwater exists, especially in rapid-infiltration wastewater basins, stormwater infiltration devices, and in agricultural areas. Nitrate has leached from fertilizers and affected groundwaters under various turf grasses in urban areas, including golf courses, parks and home lawns. Significant leaching of nitrates occurs during the cool, wet seasons. Cool temperatures reduce denitrification and ammonia volatilization, and limit microbial nitrogen immobilization and plant uptake.

The use of slow-release fertilizers is recommended in areas having potential groundwater nitrate problems. The slow-release fertilizers include urea formaldehyde (UF), methylene urea, isobutylidene diurea (IBDU), and sulfur-coated urea. Residual nitrate concentrations are highly variable in soil due to soil texture, mineralization, rainfall and irrigation patterns, organic matter content, crop yield, nitrogen fertilizer/sludge rate, denitrification, and soil compaction. Nitrate is highly soluble (>1 kg/l) and will stay in solution in the percolation water, after leaving the root zone, until it reaches the groundwater.

Pesticides

Urban pesticide contamination of groundwater can result from municipal and homeowner use of pesticides for pest control and their subsequent collection in stormwater runoff. Pesticides that have been found in urban groundwaters include: 2,4-D, 2,4,5-T, atrazine, chlordane, diazinon, ethion, malathion, methyl trithion, silvex, and simazine. Heavy repetitive use of mobile pesticides on irrigated and sandy soils likely contaminates groundwater. Fungicides and nematocides must be mobile in order to reach the target pest and hence, they generally have the highest contamination potential. Pesticide leaching depends on patterns of use, soil texture, total organic carbon content of the soil, pesticide persistence, and depth to the water table.

The greatest pesticide mobility occurs in areas with coarse-grained or sandy soils without a hardpan layer, having low clay and organic matter content and high permeability. Structural voids, which are generally found in the surface layer of finer-textured soils rich in clay, can transmit pesticides rapidly when the voids are filled with water and the adsorbing surfaces of the soil matrix are bypassed. In general, pesticides with low water solubilities, high octanol-water partitioning coefficients, and high carbon partitioning coefficients are less mobile. The slower moving pesticides have been recommended in areas of groundwater contamination concern. These include the fungicides iprodione and triadimefon, the insecticides isofenphos and chlorpyrifos and the herbicide glyphosate. The most mobile pesticides include: 2,4-D, acenaphthylene, alachlor, atrazine, cyanazine, dacthal, diazinon, dicamba, malathion, and metolachlor.

Pesticides decompose in soil and water, but the total decomposition time can range from days to years. Literature half-lives for pesticides generally apply to surface soils and do not account for the reduced microbial activity found deep in the vadose zone. Pesticides with a 30 day half life can show considerable leaching. An order-of-magnitude difference in half-life results in a five- to ten-fold difference in percolation loss. Organophosphate pesticides are less persistent than organochlorine pesticides, but they also are not strongly adsorbed by the sediment and are likely to leach into the vadose zone, and the groundwater.

Other Organics

The most commonly occurring organic compounds that have been found in urban groundwaters include phthalate esters (especially bis(2-ethylhexyl)phthalate) and phenolic compounds. Other organics more rarely found, possibly due to losses during sample collection, have included the volatiles: benzene, chloroform, methylene chloride, trichloroethylene, tetrachloroethylene, toluene, and xylene. PAHs (especially benzo(a)anthracene, chrysene, anthracene and benzo(b)fluoroanthenene) have also been found in groundwaters near industrial sites.

Groundwater contamination from organics, like from other pollutants, occurs more readily in areas with sandy soils and where the water table is near the land surface. Removal of organics from the soil and recharge water can occur by one of three methods: volatilization, sorption, and degradation. Volatilization can significantly reduce the concentrations of the most volatile compounds in groundwater, but the rate of gas transfer from the soil to the air is usually limited by the presence of soil water. Hydrophobic sorption onto soil organic matter limits the mobility of less soluble base/neutral and acid extractable compounds through organic soils and the vadose zone. Sorption is not always a permanent removal mechanism, however. Organic re-solubilization can occur during wet periods following dry periods. Many organics can be at least partially degraded by microorganisms, but others cannot. Temperature, pH, moisture content, ion exchange capacity of soil, and air availability may limit the microbial degradation potential for even the most degradable organic.

Pathogenic Microorganisms

Viruses have been detected in groundwater where stormwater recharge basins were located short distances above the aquifer. Enteric viruses are more resistant to environmental factors than enteric bacteria and they exhibit longer survival times in natural waters. They can occur in potable and marine waters in the absence of fecal coliforms. Enteroviruses are also more resistant to commonly used disinfectants than are indicator bacteria, and can occur in groundwater in the absence of indicator bacteria.

The factors that affect the survival of enteric bacteria and viruses in the soil include pH, antagonism from soil microflora, moisture content, temperature, sunlight, and organic matter. The two most important attributes of viruses that permit their long-term survival in the environment are their structure and very small size. These characteristics permit virus occlusion and protection within colloid-size particles. Viral adsorption is promoted by increasing cation concentration, decreasing pH and decreasing soluble organics. Since the movement of viruses through soil to groundwater occurs in the liquid phase and involves water movement and associated suspended virus particles, the distribution of viruses between the adsorbed and liquid phases determines the viral mass available for movement. Once the virus reaches the groundwater, it can travel laterally through the aquifer until it is either adsorbed or inactivated.

The major bacterial removal mechanisms in soil are straining at the soil surface and at intergrain contacts, sedimentation, sorption by soil particles, and inactivation. Because of their larger size than for viruses, most bacteria are, therefore, retained near the soil surface due to this straining effect. In general, enteric bacteria survive in soil between two and three months, although survival times up to five years have been documented.

Heavy Metals and Other Inorganic Compounds

Heavy metals and other inorganic compounds in stormwater of most environmental concern, from a groundwater pollution standpoint, are aluminum, arsenic, cadmium, chromium, copper, iron, lead, mercury, nickel, and zinc. However, the majority of these compounds, with the consistent exception of zinc, are mostly found associated with the particulate solids in stormwaters and are thus relatively easily removed through sedimentation practices. Filterable forms of the metals may also be removed by either sediment adsorption or are organically complexed with other particulates.

In general, studies of recharge basins receiving large metal loads found that most of the heavy metals are removed either in the basin sediment or in the vadose zone. Dissolved metal ions are removed from stormwater during infiltration mostly by adsorption onto the near-surface particles in the vadose zone, while the particulate metals are filtered out at the soil surface. Studies at recharge basins found that lead, zinc, cadmium, and copper accumulated at the soil surface with little downward movement over many years. However, nickel, chromium, and zinc concentrations have exceeded regulatory limits in the soils below a recharge area at a commercial site. Elevated groundwater heavy metal concentrations of aluminum, cadmium, copper, chromium, lead, and zinc have been found below stormwater infiltration devices where the groundwater pH has been acidic. Allowing percolation ponds to go dry between storms can be counterproductive to the removal of lead from the water during recharge. Apparently, the adsorption bonds between the sediment and the metals can be weakened during the drying period.

Similarities in water quality between runoff water and groundwater has shown that there is significant downward movement of copper and iron in sandy and loamy soils. However, arsenic, nickel, and lead did not significantly move downward through the soil to the groundwater. The exception to this was some downward movement of lead with the percolation water in sandy soils beneath stormwater recharge basins. Zinc, which is more soluble than iron, has been found in higher concentrations in groundwater than iron. The order of attenuation in the vadose zone from infiltrating stormwater is: zinc (most mobile) > lead > cadmium > manganese > copper > iron > chromium > nickel > aluminum (least mobile).

Salts

Salt applications for winter traffic safety is a common practice in many northern areas and the sodium and chloride, which are collected in the snowmelt, travel down through the vadose zone to the groundwater with little attenuation. Soil is not very effective at removing salts. Salts that are still in the percolation water after it travels through the vadose zone will contaminate the groundwater. Infiltration of stormwater has led to increases in sodium and chloride concentrations above background concentrations. Fertilizer and pesticide salts also accumulate in urban areas and can leach through the soil to the groundwater.

Studies of depth of pollutant penetration in soil have shown that sulfate and potassium concentrations decrease with depth, while sodium, calcium, bicarbonate, and chloride concentrations increase with depth. Once contamination with salts begin, the movement of salts into the groundwater can be rapid. The salt concentration may not decrease until the source of the salts is removed.

Recommendations to Protect Groundwater During Stormwater Infiltration

Table 5-1 is a summary of the pollutants found in stormwater that may cause groundwater contamination problems for various reasons. This table does not consider the risk associated with using groundwater contaminated with these pollutants. Characteristics of concern include high mobility (low sorption potential) in the vadose zone, high abundance (high concentrations and high detection frequencies) in stormwater, and high soluble fractions (small fraction associated with particulates which would have little removal potential using conventional stormwater sedimentation controls) in the stormwater.

The contamination potential is the lowest rating of the influencing factors. As an example, if no pretreatment was to be used before percolation through surface soils, the mobility and abundance criteria are most important. If a compound was mobile, but was in low abundance (such as for VOCs), then the groundwater contamination potential would be low. However, if the compound was mobile and was also in high abundance (such as for sodium chloride, in certain conditions), then the groundwater contamination would be high.

If sedimentation pretreatment was to be used before infiltration, then much of the pollutants will likely be removed before infiltration. In this case, all three influencing factors (mobility, abundance in stormwater, and soluble fraction) would be considered important. As an example, chlordane would have a low contamination potential with sedimentation pretreatment, while it would have a moderate contamination potential if no pretreatment was used. In addition, if subsurface infiltration/injection was used instead of surface percolation, the compounds would most likely be more mobile, making the abundance criteria the most important, with some regard given to the filterable fraction information for operational considerations.

Table 5-1 is only appropriate for initial estimates of contamination potential because of the simplifying assumptions made, such as the likely worst case mobility measures for sandy soils having low organic content. If the soil was clayey and had a high organic content, then most of the organic compounds would be less mobile than shown on this table. The abundance and filterable fraction information is generally applicable for warm weather stormwater runoff at residential and commercial area outfalls. The concentrations and detection frequencies would likely be greater for critical source areas (especially vehicle service areas) and critical land uses (especially manufacturing industrial areas).

Table 5-1. Groundwater contamination potential for stormwater pollutants (Pitt et al. 1996).

Categories	Compounds	Mobility (sandy/low organic soils)	Abundance in storm-water	Fraction filterable	Contamination potential for surface infilt. and no pretreatment	Contamination potential for surface infilt. with sediment-ation	Contamination potential for sub-surface inj. with minimal pretreatment
Nutrients	Nitrates	mobile	low/moderate	high	low/moderate	low/moderate	low/moderate
Pesticides	2,4-D	mobile	low	likely low	low	low	low
	γ-BHC (lindane)	intermediate	moderate	likely low	moderate	low	moderate
	malathion	mobile	low	likely low	low	low	low
	atrazine	mobile	low	likely low	low	low	low
	chlordane	intermediate	moderate	very low	moderate	low	moderate
	diazinon	mobile	low	likely low	low	low	low
Other organics	VOCs	mobile	low	very high	low	low	low
	1,3-dichloro-benzene	low	high	high	low	low	high
	anthracene	intermediate	low	moderate	low	low	low
	benzo(a) anthracene	intermediate	moderate	very low	moderate	low	moderate
	bis (2-ethylhexyl) phthalate	intermediate	moderate	likely low	moderate	low?	moderate
	butyl benzyl phthalate	low	low/moderate	moderate	low	low	low/moderate
	fluoranthene	intermediate	high	high	moderate	moderate	high
	fluorene	intermediate	low	likely low	low	low	low
	naphthalene	low/inter.	low	moderate	low	low	low
	penta-chlorophenol	intermediate	moderate	likely low	moderate	low?	moderate
	phenanthrene	intermediate	moderate	very low	moderate	low	moderate
	pyrene	intermediate	high	high	moderate	moderate	high
Pathogens	enteroviruses	mobile	likely present	high	high	high	high
	Shigella	low/inter.	likely present	moderate	low/moderate	low/moderate	high
	Pseudomonas aeruginosa	low/inter.	very high	moderate	low/moderate	low/moderate	high
	protozoa	low/inter.	likely present	moderate	low/moderate	low/moderate	high
Heavy metals	nickel	low	high	low	low	low	high
	cadmium	low	low	moderate	low	low	low
	chromium	inter./very low	moderate	very low	low/moderate	low	moderate
	lead	very low	moderate	very low	low	low	moderate
	zinc	low/very low	high	high	low	low	high
Salts	chloride	mobile	seasonally high	high	high	high	high

The stormwater pollutants of most concern (those that may have the greatest adverse impacts on groundwaters) include:

1. Nutrients: nitrate has a low to moderate groundwater contamination potential for both surface percolation and subsurface infiltration/injection practices because of its relatively low concentrations found in most stormwaters. However, if the stormwater nitrate concentration was high, then the groundwater contamination potential would also likely be high.

2. Pesticides: lindane and chlordane have moderate groundwater contamination potentials for surface percolation practices (with no pretreatment) and for subsurface injection (with minimal pretreatment). The groundwater contamination potentials for both of these compounds would likely be substantially reduced with adequate sedimentation pretreatment. Pesticides have been mostly found in urban runoff from residential areas, especially in dry-weather flows associated with landscaping irrigation runoff.

3. Other organics: 1,3-dichlorobenzene may have a high groundwater contamination potential for subsurface infiltration/injection (with minimal pretreatment). However, it would likely have a lower groundwater contamination potential for most surface percolation practices because of its relatively strong sorption to vadose zone soils. Both pyrene and fluoranthene would also likely have high groundwater contamination potentials for subsurface infiltration/injection practices, but lower contamination potentials for surface percolation practices because of their more limited mobility through the unsaturated zone (vadose zone). Others (including benzo(a)anthracene, bis (2-ethylhexyl) phthalate, pentachlorophenol, and phenanthrene) may also have moderate groundwater contamination potentials, if surface percolation with no pretreatment, or subsurface injection/infiltration is used. These compounds would have low groundwater contamination potentials if surface infiltration was used with sedimentation pretreatment. Volatile organic compounds (VOCs) may also have high groundwater contamination potentials if present in the stormwater (likely for some industrial and commercial facilities and vehicle service establishments). The other organics, especially the volatiles, are mostly found in industrial areas. The phthalates are found in all areas. The PAHs are also found in runoff from all areas, but they are in higher concentrations and occur more frequently in industrial areas.

4. Pathogens: enteroviruses likely have a high groundwater contamination potential for all percolation practices and subsurface infiltration/injection practices, depending on their presence in stormwater (likely if contaminated with sanitary sewage). Other pathogens, including Shigella, Pseudomonas aeruginosa, and various protozoa, would also have high groundwater contamination potentials if subsurface infiltration/injection practices are used without disinfection. If disinfection (especially by chlorine or ozone) is used,

then disinfection byproducts (such as trihalomethanes or ozonated bromides) would have high groundwater contamination potentials. Pathogens are most likely associated with sanitary sewage contamination of storm drainage systems, but several bacterial pathogens are commonly found in surface runoff in residential areas.

5. Heavy metals: nickel and zinc would likely have high groundwater contamination potentials if subsurface infiltration/injection was used. Chromium and lead would have moderate groundwater contamination potentials for subsurface infiltration/injection practices. All metals would likely have low groundwater contamination potentials if surface infiltration was used with sedimentation pretreatment. Zinc is mostly found in roof runoff and other areas where galvanized metal comes into contact with rainwater.

6. Salts: chloride would likely have a high groundwater contamination potential in northern areas where road salts are used for traffic safety, irrespective of the pretreatment, infiltration or percolation practice used. Salts are at their greatest concentrations in snowmelt and early spring runoff in northern areas.

It has been suggested that, with a reasonable degree of site-specific design considerations to compensate for soil characteristics, infiltration can be very effective in controlling both urban runoff quality and quantity problems (EPA 1983). This strategy encourages infiltration of urban runoff to replace the natural infiltration capacity lost through urbanization and to use the natural filtering and sorption capacity of soils to remove pollutants.

However, potential groundwater contamination through infiltration of some types of urban runoff requires some restrictions. Infiltration of urban runoff having potentially high concentrations of pollutants that may pollute groundwater requires adequate pretreatment, or the diversion of these waters away from infiltration devices. The following general guidelines for the infiltration of stormwater and other storm drainage effluent are recommended in the absence of comprehensive site-specific evaluations:

1. Dry-weather storm drainage effluent should be diverted from infiltration devices because of their probable high concentrations of soluble heavy metals, pesticides, and pathogenic microorganisms.

2. Combined sewage overflows should be diverted from infiltration devices because of their poor water quality, especially high pathogenic microorganism concentrations, and high clogging potential.

3. Snowmelt runoff should also be diverted from infiltration devices because of its potential for having high concentrations of soluble salts.

4. Runoff from manufacturing industrial areas should also be diverted from infiltration devices because of its potential for having high concentrations of soluble toxicants.

5. Construction site runoff must be diverted from stormwater infiltration devices (especially subsurface devices) because of its high SS concentrations, which would quickly clog infiltration devices.

6. Runoff from other critical source areas, such as vehicle service facilities and large parking areas, should at least receive adequate pretreatment to eliminate their groundwater contamination potential before infiltration.

7. Runoff from residential areas (the largest component of urban runoff from most cities) is generally the least polluted urban runoff flow and should be considered for infiltration. Very little treatment of residential area stormwater runoff should be needed before infiltration, especially if surface infiltration is through the use of grass swales. If subsurface infiltration (e.g., French drains, infiltration trenches, dry wells) is used, then some pretreatment may be needed, such as by using grass filter strips, or other surface filtration devices.

All other runoff should include pretreatment using sedimentation processes before infiltration, to both minimize groundwater contamination and to prolong the life of the infiltration device (if needed). This pretreatment can take the form of approaches such as grass filters, sediment sumps, and wet detention ponds depending on the runoff volume to be treated and other site specific factors. Pollution prevention can also play an important role in minimizing groundwater contamination problems, including reducing the use of galvanized metals, pesticides, and fertilizers in critical areas. The use of specialized treatment devices can also play an important role in treating runoff from critical source areas before these more contaminated flows commingle with cleaner runoff from other areas. Sophisticated treatment schemes, especially the use of chemical processes or disinfection, may not be warranted, except in special cases, especially considering the potential of forming harmful treatment by-products (such as THMs and soluble aluminum).

Most past stormwater quality monitoring has not been adequate to completely evaluate groundwater contamination potential. The following list shows the parameters that are recommended to be monitored if stormwater contamination potential needs to be considered, or infiltration devices are to be used. Other analyses are appropriate for additional monitoring objectives (such as evaluating surface water problems). In addition, all phases of urban runoff should be sampled, including stormwater runoff, dry-weather flows, and snowmelt.

- Contamination potential:
    - Nutrients (especially nitrates)
    - Salts (especially chloride)

- VOCs (if expected in the runoff, such as from manufacturing industrial or vehicle service areas, could screen for VOCs with purgable organic carbon, POC, analyses)
- Pathogens (especially enteroviruses, if possible, along with other pathogens such as Pseudomonas aeruginosa, Shigella, and pathogenic protozoa)
- Bromide and total organic carbon, TOC (to estimate disinfection by-product generation potential, if disinfection by either chlorination or ozone is being considered)
- Pesticides, in both filterable and total sample components (especially lindane and chlordane)
- Other organics, in both filterable and total sample components (especially 1,3 dichlorobenzene, pyrene, fluoranthene, benzo (a) anthracene, bis (2-ethylhexyl) phthalate, pentachlorophenol, and phenanthrene)
- Heavy metals, in both filterable and total sample components (especially chromium, lead, nickel, and zinc)

- Operational considerations:
  - Sodium, calcium, and magnesium (in order to calculate the sodium adsorption ratio to predict clogging of clay soils)
  - Suspended solids (to determine the need for sedimentation pretreatment to prevent clogging)

The Technical University of Denmark (Mikkelsen et al. 1996a and 1996b) has been involved in a series of tests to examine the effects of stormwater infiltration on soil and groundwater quality. They found that heavy metals and PAHs present little groundwater contamination threat, if surface infiltration systems are used. However, they express concern about pesticides, which are much more mobile. Squillace et al. (1996) along with Zogorski et al. (1996) presented information concerning stormwater and its potential as a source of groundwater MTBE contamination. Mull (1996) stated that traffic areas are the third most important source of groundwater contamination in Germany (after abandoned industrial sites and leaky sewers). The most important contaminants are chlorinated hydrocarbons, sulfate, organic compounds, and nitrates. Heavy metals are generally not an important groundwater contaminant because of their affinity for soils. Trauth and Xanthopoulus (1996) examined the long-term trends in groundwater quality at Karlsruhe, Germany. They found that the urban landuse is having a long-term influence on the groundwater quality. The concentration of many pollutants have increased by about 30 to 40% over 20 years. Hütter and Remmler (1996) describe a groundwater monitoring plan, including monitoring wells that were established during the construction of an infiltration trench for stormwater disposal in Dortmund, Germany. The worst case problem expected is with zinc, if the infiltration water has a pH value of 4.

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